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*Published in:*
Restoration Ecology: The New Frontier

*Chapter 16*

# RESTORATION OF MIRES

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*Restoration Ecology: The New Frontier*, Second Edition. Edited by Jelte van Andel, James Aronson.

## 16.1 INTRODUCTION

**Mires** are peatlands where peat is currently being formed (Joosten & Clarke 2002). A *peatland* is land with peat, that is, with a superficial layer of partially decomposed organic material that is at least 20–30 cm thick. Mires are ecosystems that produce peat and include *bogs* and *fens*. Bogs are mires raised above the surrounding landscape and solely fed by precipitation, especially rainwater, whereas fens are situated in depressions and, in addition to precipitation, they are also fed by ground- or surface water. Bogs are always acidic (pH $< 4.2$; Siegel *et al*. 2006), whereas fens are often base-rich or slightly acidic or neutral (pH $> 5.5$; Succow 1988; Wheeler *et al*. 2002).

Along with the term 'mires', we will also use the generic term **wetland** to describe any area of land that is sometimes or always covered by shallow water, and supports vegetation adapted to saturated soil conditions. A wetland may or may not be a peatland; it can also occur on mineral soil.

It is estimated that on a world scale the natural extent of mires was more than 4 000 000 $km^2$ (Joosten & Clarke 2002), of which 2 500 000 $km^2$ is situated in North America and Siberia (Plate 16.1). Human exploitation has altered 500 000 $km^2$ in such a way that peat accumulation has been stopped. Mire losses are highest in Europe (>50%) due to intense human population pressure and the climatic suitability for both agriculture and forestry. Losses are particularly great in western and southern Europe, where less than 10% of the natural mire area has remained (Joosten & Clarke 2002). About 80% of the mire losses here are related to agriculture and forestry. Finland and Russia in particular have drained large areas of peatland for forestry (50 000 $km^2$ and 38 000 $km^2$, respectively); this accounts for more than half of the global destruction of mires for forestry (Joosten 2009a). In most eastern European countries, between 10% and 50% of the former mires remain. In South America and Africa, peat losses are low in absolute values, but that is because these continents do not have large stocks of peat. In Africa, for instance, some 140 000 $km^2$ was originally present, but 20% of that is now already lost (Joosten 2009b). In South America, losses amount to 10%, but peat extractions are expanding, particularly in Tierra del Fuego (Argentina and Chile), where unique bogs have developed with a cover of only one *Sphagnum* species (*S. magellanicum*).

In this chapter, we discuss restoration of damaged mires worldwide, but with a particular focus on Europe. We will first describe some pristine mire types that can serve as a source of information and inspiration for restoration projects. However, they are not necessarily the appropriate **reference systems** for all **ecological restoration** projects (e.g. those in densely populated areas whose hydrological systems have often been irreversibly changed). Accordingly, we will also discuss **rehabilitation** prospects of mire remnants within strongly human-influenced **landscapes** (see also Chapter 5). The vegetation of such mire remnants may resemble that of natural mires, but **hydrologically** such systems are very different. The vegetation composition of relic mires can sometimes be used as a reference in rehabilitation projects.

## 16.2 UNDISTURBED MIRE SYSTEMS

The classification of mires is based mainly on species composition, often supported by stratigraphical and palaeo-ecological data (Brinson 1993; Grünig 1994). Succow and Joosten (2001) integrated hydrological features with the genesis of mires and linked that to nutrient status and vegetation composition. Here we focus on the following five types: rainwater-fed bogs, groundwater-fed percolation fens, terrestrialization fens, spring fens and floodplain fens.

### 16.2.1 Rainwater-fed bogs

Bogs are fed by rainwater although some peripheral parts – the so-called lagg zone – can be influenced by water from surrounding areas that has been in contact with mineral soil (Plate 16.2a). Most bog systems originate from lakes and groundwater-fed mire systems, over which a developing rainwater lens allows *Sphagnum* moss species to grow. In this way, *Sphagnum* 'escapes' the influence of groundwater and may form a domed-shape peat body that grows above the immediately surrounding area. Groundwater flow to the bog system remains essential for the functioning of the bog system by keeping water levels stable. The living top layer of a bog consists of a loosely structured layer, called *acrotelm* (Ingram 1983). It consists of living mosses (predominantly *Sphagnum* species), intermingled with phanerogams adapted to very wet and nutrient-poor conditions. There is a distinct gradient in hydraulic conductivity in this situation. The underlying, more firm and decayed peat layer called *catotelm* can store a large amount of water, thus maintaining the required wet conditions for the typical bog species. Excess water flows to the periphery of the bog through the acrotelm. In dry periods, the water level in the acrotelm drops, and reaches peat layers with much lower hydraulic conductivities. Water flow in these layers then becomes very low. This regulating feedback mechanism prevents a rapid loss of water from the bog during dry periods (Joosten 1993; Succow & Joosten 2001).

Various types of bog have been distinguished on the basis of their morphology (Glaser 1992; Glaser *et al.* 1997). 'Blanket bogs', which cover the undulating landscape like a blanket, are common in Ireland and Scotland. 'Plateau bogs' are large bogs with a flat top that used to be common in north-western Europe, but are now – as living mire systems – very rare. Still larger bog systems occur in Canada (Warner 2005), Russia (Yurkovskaya 2005), Scandinavia, the Baltic states and Belarus, as well as in Tierra del Fuego, in southernmost South America (Coronato *et al.* 2006).

### 16.2.2 Percolation fens

These groundwater-fed fens are large peat-forming gently sloping systems, sometimes semiforested, but often almost treeless, in which case they are dominated by sedges and bryophytes (Plate 16.2b). These fens occur in mountain areas and lowland river valleys, in situations where the groundwater comes from the surrounding catchment area. As in natural bogs, the uppermost peat of natural fens is loosely structured and permits a rather rapid flow of water during wet periods of the year. This type is referred to as percolation mire since, in contrast to other peatland types, a substantial part of the groundwater flow in this case occurs through the rather permeable peat in the upper layers of the mire (Succow 1988). Large natural fen systems of thousands of hectares occur in Canada (Warner 2005), Scandinavia (Lindholm & Heikkilä 2005), the Baltic States (Pakalne & Kalnina 2005), Russia (Yurkovskaya 2005), Tierra del Fuego (Coronato *et al.* 2006), Tibet (Tsuyuzaki 2006) and South Africa (Grundling & Grobler 2005). Such large peat-forming, groundwater-fed mires were also widespread in north-western Europe, some centuries ago, but at present only remnants of percolating fens occur there, which no longer form peat (Succow & Joosten 2001). In the best-preserved fen systems, such as those in the Biebrza Valley, Poland, typical fen species co-occur with a large number of fen meadow species that mark the transition towards grasslands. Locally peat formation can occur, but many such fen systems no longer accumulate peat, and in most systems regarded as fens the peat is actually degrading. One of the largest fen systems in the world, the Everglades in Florida, United States, is a large calcareous fen that originally occupied more than one million hectares (Kadlec 2009). At present more than half of it has been reclaimed for agriculture (Richardson 2008). In section 16.4.2 we will discuss attempts to restore damaged parts of the Everglades.

### 16.2.3 Terrestrialization fens

Terrestrialization fens occur almost everywhere where peat formation can occur, and they are abundant under natural conditions. In human-dominated environments, however, most terrestrialization mires are artificial. In the lowlands of north-western Europe, for instance, the natural mires (fens and bogs) are long gone (Plate 16.2a and 16.2c) and secondary terrestrialization has taken place in former peat extraction sites. Most of these newly formed terrestrializing mires ('floating mats') are small, ranging in size from less than 1 ha to a few hundred hectares, such as the Norfolk Broads in England and the Dutch Weerribben or Wieden (van Wirdum *et al.* 1992; van Diggelen *et al.* 1996). In the Mississippi River delta plain in Louisiana, United States, there is a very large, floating freshwater system of circa 3000 ha (Sasser & Gosselink 1984).

Terrestrialization mires are ecologically very diverse. Eutrophic plant communities, such as reeds, form floating rafts on which in later stages mesotrophic or even oligotrophic plant communities can develop when the peat layer is sufficiently thick to allow the formation of rainwater lenses (van Wirdum *et al.* 1992). When groundwater supply is insufficient or the surface water contains too many nutrients and sulphates, these terrestrializing fens are rapidly overgrown by shrubs and acidify rather quickly (van Diggelen *et al.* 1996).

### 16.2.4 Spring fens

Spring fens are almost exclusively fed by groundwater discharge (Plate 16.2c). They generally occur in landscapes with rather complex geological and hydrological conditions. Usually they occur in places where groundwater flowing through an artesian aquifer bursts through a rupture in the overlying confining layer and discharges at the land surface. Such a rupture is often the result of changes in the original stratification of sediments (e.g. where tectonic movements have occurred) (Wilcox *et al.* 1986; Grootjans *et al.* 2005). Spring systems can be found as elevated spots, in lowlands in the middle of a sloping landscape or even close to a hilltop. In all these settings, spring fens occur at the inflection point where local groundwater flow systems intersect the land surface (Almendinger & Leete 1998).

In calcareous substrate areas, the regular discharge of groundwater may accumulate large amounts of $CaCO_3$ (*travertine*) up to a height of 30 m (Grootjans *et al.* 2005; Pentecost 2005). When groundwater, supersaturated with $CaCO_3$, comes into contact with the atmosphere, $CO_2$ escapes and $CaCO_3$ precipitates. Spring mire plants, and mosses in particular, stimulate this process by using the $CO_2$ in the discharging groundwater as a source of carbon. The result is that $CaCO_3$ precipitates on plant leaves, and when they die off and decompose, $CaCO_3$ is deposited. Under favourable climatic conditions, peat formation may occur. Peat formation in spring mires is sensitive to changes in landscape hydrology and climatic conditions that trigger changes in *groundwater discharge* leading to the *decomposition* of the peat layers. Strongly decomposed peat has a high resistance to water flow, thus blocking water transport in the spring mound. The discharging groundwater will then force its way out somewhere else in the spring system, thereby creating new opportunities for peat or travertine formation (Wolejko *et al.* 1994). Some of the best-preserved spring-mire systems anywhere can be found at the base of the Alps and the Slovakian Tatra Mountains, and in Latvia (Pakalne & Kalnina 2005). Outside Europe, spring mires occur in the United States (Komor 1994; Glaser *et al.* 2004; Middleton *et al.* 2006), South Africa (Scott & Vogel 1983), Siberia (Schipper *et al.* 2007) and Australia (Whinam & Hope 2005).

Most spring mires in western Europe have disappeared due to decreased inflow of groundwater from surrounding areas as a result of changes in land use. In north-western Poland, however, many spring systems are still present and their infiltration areas have not been hydrologically disturbed. Yet most of them have been damaged by drainage in downstream lake areas (Plate 16.3c; Wolejko *et al.* 1994). Decreasing lake levels in the nineteenth century have triggered erosional processes that have washed away most of the peat that has accumulated around these spring systems.

### 16.2.5 Floodplain fens

In hydrological terms, floodplain fens (Plate 16.2d) are very dynamic systems. In winter and spring, intensive flooding may deposit large amounts of sand, silt or clay. In summer, water tables may drop to 1 meter or more below the soil surface. Under such conditions, nutrient availability can be very high (Olde Venterink *et al.* 2003) resulting in high productivity of floodplain vegetation. On a geological time scale, rivers frequently change their course, leading in turn to modified flooding frequencies and sedimentation rates. Eutrophic floodplains may change into mesotrophic fens and existing fens, or even bogs, may become flooded with surface water. Examples of large floodplain mires in Europe include the Danube floodplain in Romania (Vadineanu 2009), the Oder floodplains on the border of Poland and Germany, the Narev floodplain in Poland and the Rhône River delta in southern France (see also Chapter 17). Large floodplain mires outside Europe are the Okavanko Delta in Botswana (McCarthy & Ellery 1998) and the Mesopotamian marshes in Iraq (Maltby 2009).

## 16.3 CAUSES OF DISTURBANCE OF NATURAL MIRES

The main causes of destruction of and disturbance in natural mire systems vary from direct peat extraction for fuel to indirect changes in landscape hydrology and atmospheric nitrogen deposition due to changes in *land use* by agriculture, forestry or urbanization. The indirect hydrological changes and their effect on nutrient availability in the peatland may be subtle, but can lead to natural mires changing into systems that no longer accumulate peat, but instead actually release carbon into the atmosphere. The change from natural mires to disturbed ecosystems is illustrated in Plate 16.3, which shows typical remnants of natural mire types after large changes in land use during the past few centuries. The time period in which such land use changes began may differ greatly, but the outcome is usually similar.

### 16.3.1 Peat extraction

Peat extraction is responsible for circa 10% of global mire losses, with new peat extraction sites being established at a rate of 10 $km^2$ annually (Joosten & Clarke 2002). Countries with large peat extraction activities for energy are Finland, Russia and Ireland. In Ireland, in particular, this use of peat for energy is criticized by national and international organizations because burning peat is very inefficient as compared to burning coal or gas. In other countries, like Germany, the Baltic states and Canada, peat is primarily extracted as a raw material for horticultural growing media. Even when peatlands are no longer used for peat extraction, the peat losses continue. Deeply drained peatlands suffer from peat loss through shrinkage and mineralization (Oleszczuk *et al.* 2008). This process may cause losses as great as 2–3 $cm\,yr^{-1}$ (Joosten 2009a) and trigger the release of enormous amounts of greenhouse gases into the atmosphere (ca. 40 kg of $N_2O$ $ha^{-1}yr^{-1}$ and 10 t $ha^{-1}yr^{-1}$ of $CO_2$; Armentano & Menges 1986). Altogether, these processes have significant impacts on global greenhouse gas cycles. On a global scale, peat losses due to agriculture, forestry and wildfires amount to circa 500 million tons of carbon per year (Joosten 2009b, see also Box 16.1 on Indonesian peatlands). In contrast, global peat accumulation is estimated to at least five times less than that. This makes it clear that global peatlands have switched from being carbon sinks to carbon emission sources (Couwenberg *et al.* 2010).

### 16.3.2 Indirect changes in hydrological regimes

Most fens and bogs in north-western Europe have been converted into intensively used agricultural land or commercial forest plantations. These land use changes have altered groundwater flow patterns and groundwater composition, which of course has negative consequences for remaining fens and bogs. Ecohydrological research in Poland and the Netherlands has shown that large drainage works in a landscape have fragmented the former regional flow systems into smaller local cells. This fragmentation of hydrological systems has resulted in significant hydrochemical differences, which can be observed even within a single small fen (Wassen *et al.* 1996). In the Biebrza catchment in eastern Poland, for example, the hydrochemical and associated trophic gradients in the vegetation are still very smooth and gradual over large distances (Wassen *et al.* 1996). This relative spatial uniformity is related to slow but continous flow of groundwater in the mire that keeps it saturated with mineral-rich groundwater. Fen systems in the Netherlands resemble Polish fens with respect to vegetation, but their hydrological systems are much more variable. In areas with a large precipitation surplus, drainage promotes **acidification** in peatlands because calcium- and iron-rich groundwater is replaced by rainwater (Schot *et al.* 2004). Such processes can even trigger bog formation, when sufficient precipitation water is present.

In order to compensate for water losses due to drainage, surface water is often supplied to agricultural areas during summer. This artificial water supply tends to reverse the natural fluctuation pattern such that low water levels occur in winter and spring and high water levels occur in summer. Most of this water originates from large rivers with much higher nutrient and sulphate content than groundwater. A higher nutrient availability in the water leads to **eutrophication**. But in organic soils, high sulphate levels may also trigger eutrophication. This process is called *internal eutrophication*. Under anaerobic conditions, the availability of alternative electron acceptors, such as sulphate, may strongly stimulate the breakdown of organic matter and increase the availability of nutrients (Smolders *et al.* 2006). During sulphate reduction, sulphide is produced, which in relatively high concentrations is toxic for most vascular plants (Lamers *et al.* 2002). In most groundwater-fed fen soils, sulphide will not reach toxic concentrations since it is chemically bound by iron. When sulphide production, however, exceeds the availability of iron, free sulphide concentrations increase (Caraco *et al.* 1989). When iron availability in the soil becomes limited, for instance when groundwater supply to the fen has stopped, phosphate concentrations in the pore water can increase to high levels. Such changes in nutrient cycling can be very harmful for phosphorus-limited fens (Koerselman & Verhoeven 1995; Richardson 2008). In general, anoxic iron-rich groundwater tends to slow down nutrient cycling,

### Box 16.1 Peatlands in Kalimantan: Destruction and hopes for restoration

Although precise data are absent, it is estimated that Indonesia may have 27 Mha of peatland, the largest area found in any tropical nation. In 1995, the then president Suharto initiated the so-called 'Mega-Rice project', which was intended to drain one million hectares of peatland in central Kalimantan for rice production. The aim was to produce two million tons of rice per year in order to keep Indonesia self-sufficient in rice production and at the same time redistribute people from the over-populated islands (Java, Bali, Lombok; each with around 700 people $km^{-2}$) to less populated areas, such as Kalimantan. The total area of the mega-project was almost 1.5 Mha, including 0.9 Mha of peat swamps, which represents 41% of the total peat swamps in central Kalimantan. In order to make the area suitable for rice production, primary tropical forest had to be cleared and an extensive drainage system had to be dug. The main drainage channel was 110 km long, 25 m wide and 6 m deep. The secondary, tertiary and quaternary channels had a total length of more than 4500 km. The canals were dug without sufficient knowledge of the hydromorphological properties of peatlands. Many peatlands were drained too deep, resulting in subsidence, peat oxidation and acidification, and greatly increased risk of wildfire. Indeed, the biomass residues from logging activities provided fuel for immense, uncontrollable fires in 1997 and, to a smaller extent, in consecutive years. More than 500 000 ha of land was intentionally or accidentally burned in 1997. Many fires were lit by companies and farmers as a cheap way of clearing land for agriculture, especially oil palm and rubber plantations. In the affected areas, the surface-drained peat that was supposed to support crops, burned away. In 1997, more than 40 000 people suffered from respiratory diseases, and over 1000 people died, from smog-related illnesses. Neighbouring countries including Malaysia, Singapore, Brunei and even Thailand, suffered the consequences as well. Schools were closed, airline flights cancelled and thousands of people wore face masks for weeks on end. Environmental costs were also extremely high. Page *et al*. (2002) estimated that in 1997 alone 810–2570 Mt C was released to the atmosphere as a result of burning peat and vegetation. This is equivalent to 13–40% of the mean annual global carbon emissions from fossil fuels. The haze and fires during August–December 1997 cost the region an estimated US$4.4 billion. Within two years, it became apparent that the project would never achieve its objectives and the new president of Indonesia, President B.J. Habibie, reversed the decision of the former president Suharto, and stopped the project. By then around US$500 million had been invested, but little had been achieved. Most of the forest, however, had been destroyed. Habibie soon decided to convert the peat areas into agricultural land to grow other food crops than rice and also to further development of oil palm and rubber plantations. Although the new plans stress the need for wise-use principles for using peatlands, the main emphasis was and still is on economic development. The presidential decree was criticized by environmental groups because it lacked consideration of the ecological functioning of tropical peatlands and ignored their environmental values. Areas that were designated for peat and wildlife protection in this new policy were highly fragmented and therefore not suitable for conservation objectives as they are situated too close to extensively drained development areas. Most importantly, carrying out these new plans did not stop the fires. In the last decade; new, and somewhat more integrated plans have been made to restore the area, paying more attention to conservation and integral water management. However, little actual implementation has yet taken place. $CO_2$ emissions from drained peatlands in Indonesia still amount to more than 500 Mton $CO_2$ per year, and this excludes emissions due to fires. The emissions from Indonesian peatlands are almost half of the global $CO_2$ emissions per year from drained peatlands (Joosten 2009b). European coutries and Australia are now investing many millions of dollars in projects that aim to block the drainage channels and rewet the peat again, but the successes are still very modest.

Based in large part on Rieley (1999) and Joosten (2009b).

while oxic surface water tends to stimulate nutrient cycling. For further reading on the effects of oxidizing effects of nitrate- and sulphate-rich water on internal eutrophication in wetlands, we refer to Smolders *et al*. (2010).

### 16.3.3 Atmospheric N deposition

Agricultural areas with intensive cattle breeding emit high amounts of $NH_3$ into the atmosphere, which also reaches nature areas through dust particles and precipitation (*external eutrophication*). Atmospheric nitrogen and sulphur deposition have a very negative effect on almost all low-productivity wetland ecosystems. The concept of critical N load has been developed to indicate the annual amount of N deposition above which there is a realistic risk that essential functions of an ecosystem may be impaired (Achermann & Bobbink 2003). *Critical nitrogen loads* for bogs are 5–10 kg of N $ha^{-1}yr^{-1}$, and for mesotrophic (unmanaged) terrestrialization fens they are 20–35 kg (Bobbink *et al*. 1998). When the deposition of N increases, the *Sphagnum* species start to accumulate N in amino acids, or mineral N is no longer absorbed and becomes available for vascular plants such as *Molinia caerulea* and *Betula pubescens* (Lamers *et al*. 2000).

## 16.4 RESTORATION APPROACHES, SUCCESSES AND FAILURES

There is a growing awareness that *rewetting* (i.e. increasing water levels) in drained peatlands not only is beneficial for protecting **biodiversity** but also reduces greenhouse gas emissions. New economic uses of peatlands, such as biomass cultivation for biofuel, are now emerging, since many peatlands have become unsuitable for modern agricultural production (Pfadenhauer & Grootjans 1999). In many peat areas, in Europe in particular, the maintenance costs and subsidies paid by taxpayers are simply too high, and the revenues too small, to allow this type of land use on peat soils to continue.

For biodiversity conservation and other environmental objectives, rewetting should preferably be done with anoxic, unpolluted groundwater, because this will conserve the peat and prevent further emissions much more than rewetting with surface water (Smolders *et al*. 2010). Rewetting can be achieved by constructing dams or filling in drainage ditches. Terminating groundwater abstraction activities will also lead to rewetting.

When restoration to the natural state is no longer possible, **rehabilitation** of some wetland functions is a more appropriate and realistic goal (e.g. Wheeler & Shaw 1995). When a wetland is damaged beyond a certain **threshold** of repair, rehabilitation efforts may lead to remodelling the system towards a state that probably never existed before, but is far preferable to sheer abandonment. In areas that have been abandoned by farmers, industries or other land users, most nature development projects now in fact explicitly aim at designing and creating an entirely new wetland ecosystem.

In what follows, we will discuss restoration experiences, and lessons learned, in western Europe and elsewhere, with a focus on damaged nature reserves or abandoned peat extraction areas.

### 16.4.1 Restoration of rainwater-fed bogs

In north-western Europe and in North America, many restoration and rehabilitation efforts include attempts to restart peat growth in large bog complexes where peat has been extracted for commercial use (Wheeler *et al*. 2002; Money *et al*. 2009). Reinitiation of *Sphagnum* growth in such flat surfaces has proven to be very difficult (Vermeer & Joosten 1992). Essential for successful bog restoration is to re-create a new functional acrotelm (Money *et al*. 2009). Without that, peat formation cannot occur. So, the first step is to create good conditions for very active *Sphagnum* growth. In degraded bog remnants, these conditions are unfavavourable due to (1) too-high water table fluctuations (>25 cm), (2) high atmospheric N deposition and (3) low dissolved inorganic carbon (DIC) concentrations in the pore water.

#### Too-high water table fluctuations

Water table fluctuations in degraded bog remnants are usually too large for *Sphagnum* growth, due to the too-low storage capacity (specific yield) of the remaining peat layers and because of water losses to surrounding (drained) agricultural areas. Restoration measures in such cases usually consist of increasing water levels in the bog to over the surface by building large dams in and around the bogs. Water table fluctuations can be largely reduced by such measures, but large parts of the bog may become permanently inundated, which can limit *Sphagnum* growth due to light limitation and lack of carbon dioxide. Humic acids from the remaining peat substrate lead to a condition known as dystrophic (i.e. water that is poor in nutrients and contains a high concentration of humic acid, with limited penetration of light). Light limitation can thus hamper the growth of submerged *Sphagnum* when water depth exceeds 0.5 m (Wheeler & Shaw 1995). Shallow flooding, however, often results in strong fluctuations of the water level and the drying out of the topsoil in dry years (Price 1998). Increasing groundwater levels in surrounding agricultural areas will improve the restoration success in the bog remnants considerably, but this is an option only when these drained areas can be purchased, or otherwise compensation can be arranged for the relevant farmers or landowners.

Experimental research has shown that good restoration results can be achieved when the peat either swells up to the newly created water table or becomes buoyant, in both cases creating a favourable substrate for *Sphagnum* mosses. Floating of *Sphagnum* vegetation is usually caused by high methane production from underlying substrates (Lamers *et al*. 2002; Tomassen *et al*. 2003). Buoyancy of *Sphagnum* mats is stimulated when poorly humified *Sphagnum* peat is still present (Tomassen *et al*. 2004), or when slightly or somewhat calcareous groundwater enters the peat base, thus stimulating decomposition (Malmer & Wallén 1993; Smolders *et al*. 2003). Some field experiments are presently underway in which small amounts of lime are added to the peat before rewetting and less degraded peat from other areas is added in order to stimulate decompostion processes and methane production (van Duinen *et al*. 2011).

### High atmospheric N deposition

Limpens *et al*. (2003) showed that high atmospheric N deposition (30–35 kg of N $ha^{-1}yr^{-1}$) reduces the growth of rainwater-fed *Sphagnum* species and favours the growth of *Molinia caerulea* and *Betula pubescens*. Above 18 kg of N $ha^{-1}yr^{-1}$ *Sphagnum* species cannot take up all the $NH_4^+$ from the soil solution and the nitrogen becomes available for vascular plants, which start to shade the *Sphagnum* plants, thereby reducing their growth considerably.

### Low dissolved organic carbon concentrations

Initial stages of bog formation usually consist of a dense vegetation of submerged or floating *Sphagum* species, like *Sphagnum cuspidatum* or *S. fallax*. Carbon dioxide availability is crucial to the growth of submerged *Sphagnum* species (Smolders *et al*. 2001). In general, such species are only able to reach buoyancy when carbon dioxide concentrations in the pore water (i.e. the water filling the spaces between grains of sediment) are higher than 500 μmol $l^{-1}$ (Smolders *et al*. 2003). Carbon dioxide in bogs is derived mainly from decomposition processes. As a result, carbon dioxide concentrations are highest just above the inundated peat soils. Therefore, shallower waters will present not only more favourable light conditions but also higher carbon dioxide concentrations for submerged *Sphagnum*. Once submerged or floating *Sphagnum* species have established, a structure may be formed on which true bog species can establish and form a new functional acrotelm. Key species for acrotelm development are *Sphagnum magellanicum*, *S. papillosum*, *S. imbricatum*, *S. fuscum* and *S. rubellum*. These typical 'hummock' and 'lawn' species are usually very slow colonizers compared to the wetter 'hollow' species such as *S. cuspidatum* and *S. fallax*. Introduction of key species in carpets dominated by hollow species or on bare substrates appears to be very successful, indicating that the main constraint is colonization (Smolders *et al*. 2003).

Several recent studies suggest that in the first stages of peat regeneration, *Sphagnum* plants may also need support from tussock-forming species, such as *Eriophorum vaginatum*, or from perennial species such as relatively tall *Carex* species that can form many erect shoots, without shading the *Sphagnum* plant too much (Ramseier *et al*. 2009). In Canada, mulching with straw has been used for protection against evapotranspiration losses and to provide a support structure for young *Sphagnum* plants (Rochefort *et al*. 1995).

## 16.4.2 Rehabilitation of groundwater-fed fens

Repairing the hydrological regime of a wetland is more complex than just raising water levels. In groundwater-fed fens, in particular, increasing water levels may lead to **acidification** when the discharge of base-rich groundwater cannot be reinstated. Therefore, successful management of groundwater-fed fens should be approached at a broader scale that includes the landscape-scale management of groundwater systems. Not only quantitative aspects are important but also qualitative aspects. Leaching of nitrate to the groundwater, for instance, can cause large-scale mobilization of sulphate from geological pyrite or gypsum deposits and the immobilization of ferrous iron (Smolders *et al*. 2010). A *landscape-scale approach* often implies changing land use in the entire surrounding catchment area. This requires stakeholder participation. Without that support, fen rehabilitation is almost unachievable because of conflicting land use claims from farmers, foresters, city dwellers and so on (Klimkowska *et al*. 2010b).

One of the largest rehabilitation projects in the world is the well-known project in the Florida Everglades (Kadlec 2009), where large-scale hydrological measures are being executed to reinstate insofar as possible, the original flow patterns of this large fen system. Much effort is given to reduce phosphorus loading to the protected fen areas of the Everglades National Park. A substantial amount of phosphorus stems from agriculture, and therefore this sector should logically contribute substantially to the cost of rehabilitation. If farmers do not meet the goals of phosphate reductions, as formulated in *Best Management Practices*, they are required to pay extra taxes to the state of Florida. The total cost of the actions intended to maintain and restore the ecological character of the Everglades is currently estimated at more than US$10 billion, and the figure is rising (Kadlec 2009).

### 16.4.3 Restoration of terrestrialization fens

In western Europe, terrestrialization fens in lake areas are often the last refuge for nutrient-poor fen plants that were once widespread in natural groundwater-fed fens. With increasing eutrophication of surface waters, the lakes also became polluted and, as a consequence, nutrient-poor fen species in the terrestrialization fens have become rare (van Wirdum *et al*. 1992). Restoring such nutrient-poor fens has to start with improving the water quality of the lakes in which the fens occur. Technical solutions, such as sewage treatment plants, can decrease nutrient concentrations. In densely populated areas, however, sewage treatment may not be suffcient, since nutrients from diffuse sources may enter a protected nature area via canals or via groundwater flows from surrounding agricultural areas. A proper way to deal with the complex relationships in the water systems and human exploitation of water (e.g. for drinking water, navigation and tourism) is integral water management. In other words, all the interrelated hydrological problems in a watershed – including pollution of underwater sediments, local and regional input of pollutants, shoreline management, water level management, groundwater extraction for drinking water and industrial use – should ideally be addressed simultaneously and in an integrated fashion.

Not all terrestrialization fens need management. In oligotrophic lakes, small bog hummocks or low-productive fen vegetation can develop soon after the formation of a floating mat of reed or large sedges. In eutrophic lakes, terrestrialization can lead to monocultures of reed (*Phragmites australis*) with little value for nature conservation. In such cases, cutting can reduce the vitality of the reed and increase biodiversity. Summer mowing reduces the vitality of *Phragmites* more than winter mowing (van Diggelen *et al*. 1996). Excavating new peat ponds is a measure to restart the terrestrialization process to provide new habitats for rare and endangered fen species (Beltman *et al*. 2001). Excavation should be practised only when the new peat ponds can be supplied with very clean water, preferably groundwater.

### 16.4.4 Rehabilitation of spring mires

Most degraded spring-mire systems can no longer be restored since the original peats and sediments have disappeared and the water discharge sites have shifted to lower sections of the spring system and cannot be directed upwards. The replacement communities, consisting of plant and animal species adapted to low temperatures and constantly flowing water, are also highly endangered in western Europe (Wolejko *et al*. 1994). Rehabilitation measures should aim at stabilizing water outflow in springs and preventing pollution in infiltration areas. When the discharge of groundwater has diminished or the spring water has been polluted in surrounding infiltration areas, the most obvious measure is to protect the direct catchment areas and abolish drainage and fertilization practices. In efforts to rehabilitate a severely damaged spring mire complex (6 ha) in the Sernitz region of north-eastern Germany (Koska & Stegmann 2001), several measures were taken concurrently to raise the water table. Measures included (1) constructing a series of wooden dams in the largest drainage ditches, (2) completely filling ditches with peat, (3) reflooding parts of the mire with spring water and (4) perforating the impervious gyttja layers at the base of the mire. The best results were obtained with complete filling of drainage ditches. Flooding parts of the mire with spring water that had accumulated behind wooden dams could not prevent severe desiccation during summer. The series of wooden dams in the main drainage ditch were also ineffective, since they only slowed down drainage, rather than preventing. Their impact remained very local. Perforating the impervious layers to enable spring water to discharge at the top of the spring mire also had a very limited effect and did not contribute to the rewetting of the mire.

### 16.4.5 Restoration of floodplains

Stimulating flooding in eutrophic flood mires can be an effective measure to combat acidification and can, under certain conditions, compensate for the influence of drainage (see also Chapter 17). Nutrient levels and sulphate levels should be very low to prevent rapid eutrophication. Remnants of former extensive flood mires are often surrounded by agricultural areas with much lower water levels. Former discharge areas have turned into infiltration areas and **acidification** of the topsoil has degraded the fens and fen meadows to the extent that iron levels have become critically low (van Duren *et al*. 1998; Lamers *et al*. 2002). Reflooding with surface water, even when pre-purified by **helophyte filters** (reed swamps that reduce the nutrient content of the surface water), does not always restore the iron concentrations to higher levels, since surface water usually contains very little iron. If iron is fixed as $FeS_2$ (*pyrite*), flooding with surface water may lead to a higher pH during winter and spring, but during summer, when groundwater levels drop, the topsoil acidifies again due to oxidation of $FeS_2$ (Lamers *et al*. 2002). Permanent flooding could prevent acidification, but will also increase the availability of nutrients in the soil, leading to very productive plant communities (Lucassen *et al*. 2004). Such restored wetlands, however, regain the capability to capture nutrients from the surface water system (Olde Venterink *et al*. 2003; Richardson & Hussain 2006), and thus provide habitats for large populations of waterfowl (Middleton *et al*. 2006). Flooding is also important for the redistribution of seeds within restoration sites (Wheeler *et al*. 2002; Jansen *et al*. 2004). Flooding does not automatically restore peat formation. Many species that respond positively on renewed flooding, such as *Glyceria maxima*, *Acorus calamus* and *Phalaris arundinacea*, do not form peat. For peat formation, the vegetation should consist of wetland species with stiff or fibrous tissues that degrade slowly, such as *Phragmites australis* and various tall sedges (*Carex* spp.; Richert *et al*. 2000).

## 16.5 PERSPECTIVES

In densely populated areas, **ecological restoration** of mires is almost impossible since hydrological conditions of natural mires are usually dependent on the larger hydrological systems of the surrounding landscape. If restoration implies restrictions to regional water use, support from society will often be lacking entirely. Targets for restoration projects should therefore be set carefully and clearly in order to obtain political support (Swart *et al*. 2001). In practice, mire ecosystems can only be restored on a very local scale, and although features of natural mires may return, the new ecosystems will differ from the ones that have been destroyed in former times. Successful projects have in general been executed at sites little affected by intensive agriculture and drainage.

Successful restoration in densely populated areas often implies repairing damage at high costs. From an economic perspective, conservation of still-existing undisturbed ecosystems is much more cost-effective. However, there are increasing numbers of situations where pressures and motivations are sufficient to trigger – and economically justify – mire restoration elsewhere. Failures to repair damaged elements of mires are as numerous as the successes, but they are usually not well documented. Failures are often caused by an incorrect diagnosis of the restoration prospects of the site (Grootjans *et al*. 2006), lack of knowledge on ecological processes affecting the site negatively, and expectations that are too high or unrealistic.

Reduction of the amount of N deposition remains a prerequisite for successful restoration of many nutrient-poor ecosystems, such as bogs, fens and several types of fen meadow. In some European countries, atmospheric N and $SO_2$ deposition has declined considerably during the past 20 years, due to more strict environmental legislation. Freshwater ecosystems have benefited considerably from drastic reductions of phosphorus emissions in the surface water, but in many streams and rivers the amount of sulphate is still much too high and causes eutrophication in terrestrial mire systems when they are flooded.

Additionally, leaching of nitrate to the groundwater should receive much more attention, especially for its potentially large-scale mobilization of sulphate from geological pyrite deposits and the immobilization of ferrous iron. Many water managers still ignore that sulphate-induced eutrophication and sulphide toxicity can strongly affect the biodiversity of wetlands that are directly – or indirectly, via surface water – fed by sulphate-rich groundwater (Lamers *et al.* 2002; Smolders *et al.* 2006). The reduction of nitrogen loads to the groundwater should be a major objective in programmes designed to tackle this problem.

Happily, the prospects for **rehabilitation** of damaged ecosystems are relatively good in most developed countries, since (1) much experience is now available to repair or even rebuild damaged ecosystems, (2) due to new legislation some environmental stress factors, such as high nutrient loads in surface waters and atmospheric nitrogen deposition, are decreasing and (3) public and political support for restoration is increasing. In developing countries, by contrast, the situation is less clear. Since environmental laws are less strict or less strictly controlled, environmental problems may be imported from the West. Public and political support for restoration activities is largely lacking. However, the quality and the number of relatively undisturbed wetlands are still very high in many parts of the world, and the costs to repair the damage in such systems are still low. It could be very beneficial to develop projects that combine economic development and maintenance of (semi)natural mires and wetlands (Wichtman & Succow 2001). With modern technology, mires can be destroyed relatively easily. But in most cases the economic benefits of such actions are temporary, because once destroyed, mires can no longer supply services to society. Societal costs for restoration of damaged mires are very high, and very often the damage is so severe that full restoration is no longer possible. To summarize the prospects of mires, we could say: the future of mires is in conservation.

## ACKNOWLEDGEMENTS

The authors wish to thank Paul Glaser, Jelte van Andel and James Aronson for their helpful comments on an earlier version of the manuscript.